Ecological corridors also operate in an urban matrix: A test case with garden shrews

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Abstract The fragmentation of landscapes, induced by the growing urbanisation, is a major cause of biodiversity loss. Ecological corridors are landscape elements that make up for the negative effects of habitat fragmentation in agricultural or natural landscapes. However, their effectiveness have been rarely assess in an urban context. To do so, we analysed shrews in woodlots, corridors and domestic gardens that are connected or disconnected from a corridor. Indeed, these mammals are very sensitive to fragmentation and could play an important ecological role even in small and isolated areas such as domestic gardens. Ninety-seven shrews, from three species were trapped. In our study, *Sorex coronatus*, *Sorex minutus* and *Crocidura russula* were not widely distributed and mainly occurred in woodlots, corridors and connected gardens. In gardens, shrew occurrences were mainly determined by landscape configuration with negative effects of the distance to the corridor and the distance to the woodlot, stronger for *C.russula* than *Sorex* species. At the local scale, garden management and vegetation showed a positive effect mainly for *Sorex* species. The quality of gardens seems good enough to permit the co-occurrence of several species of shrews and thus competition was not identified as a major process for determining the occurrence. We showed that, as in other ecosystems, ecological corridors in urban context can play an effective role for the distribution of organisms with low dispersal capabilities and should be develop in planning strategies.

Keywords Urban ecology · Domestic garden · Fragmented landscape · Corridors · Shrew · Corridor

Introduction

The present and future expansion of cities (3 % of the earth surface to 6 % in 2030, World Bank 2009) results in the replacement of natural or rural local habitats by more artificial structures like

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buildings and roads (Pauchard et al. 2006). This change of land use is referred to urbanisation which is a major driver of local habitat loss and fragmentation (Collinge 1996; McDonnell et al. 1997; Porter et al. 2001; European Environment Agency 2011). In the patch-matrix model (Forman and Godron 1981), fragmentation transforms large patches of habitat into smaller and more isolated patches embedded in a hostile matrix (McGarigal and Cushman 2002). These dramatic changes of landscape configuration have consequences on many levels; from populations to the overall functioning of ecosystems (Saunders et al. 1991; Fischer and Lindenmayer 2007). At the population level, fragmentation caused a brutal subdivision of one population in smaller subsets located in patches sometimes too small to support viable subpopulations. These subpopulations could go instinct due to environmental or demographic stochasticity. Fragmentation also altered the fundamental process of dispersal (Clobert 2001). Patches are generally isolated in a hostile matrix which limits the dispersal of individuals and genes between patches. As a consequence, the probability of recolonisation by other subpopulations is affected (Hanski 1998). At the community level, fragmentation is a key driver of 'global species lost' (Fischer and Lindenmayer 2007). By modifying the interactions between species, fragmentation modifies wildlife and the overall functioning of ecosystems (Massol et al. 2011).

In urban landscapes, patches of woodlands remain, sometimes called green spaces (Smith et al. 2006). They are generally highly fragmented (McDonnell et al. 1997) and isolated in a hostile matrix that strongly limits the dispersal of many organisms (Coffin 2007; Rico et al. 2007). Some green spaces are so isolated that some authors considered them as islands in grey oceans (Davis and Glick 1978; Clergeau et al. 2004). As a consequence, the biodiversity in green spaces is often limited to organisms with high dispersal capabilities (Croci et al. 2008; McKinney 2008; Pellissier et al. 2012). Actually, the conservation of urban biodiversity 'where people live and work' (Miller and Hobbs 2002) is an important concern because more and more landscape will become urban. Moreover, urban biodiversity could provide many services that can improve the wellbeing of urban dwellers (reviewed by Dearborn and Kark 2010).

In other fragmented landscapes, corridors—linear pieces of land which differs from the surrounded matrix (Forman 1995)—can provide the 'conduit function' by improving the movement of organisms between patches, especially those with low dispersal capabilities (Hess and Fischer 2001; Bailey 2007; Gilbert-Norton et al. 2010). These results have been integrated into landscape management policies that identify the need to protect and develop corridors into green frameworks, from local to regional scales (Jongman et al. 2004; Bryant 2006; European Environment Agency 2011). As proposed by several authors (Savard et al. 2000; Ahern 2007) and recent policies (COMOP 2009), such a strategy should be applied to urban landscapes to limit the negative effects of fragmentation on biodiversity. Associated with woodlots, corridors (here woody corridors) could be the major axes of ecological functioning in towns. They could link urban green spaces between themselves and/or with habitats located outside cities (Clergeau et al. 1998; Bryant 2006; Clergeau et al. 2006). In urban landscapes, many infrastructures like greenways, woody roadway verges or woody railway edges have a corridor structure (Ahern 2007). Even if these infrastructures have a corridor structure that cannot guarantee that they provide the 'conduit function' (Hess and Fischer 2001; Horskins et al. 2006). Moreover, some features proper to urban corridors could strongly affect their functioning like their management or the properties of the matrix. These potential corridors have a specific management compared to rural hedgerows (Henein and Merriam 1990) and are rounded by a matrix

(Ricketts 2001) also specific to the urban context, properties that could affect their functioning. The knowledge achieved in other landscapes cannot be directly applied in urban landscapes. Few studies had clearly demonstrated their positive effects on animal biodiversity (Tigas et al. 2002 on bobcats and coyotes, Vergnes et al. 2012 on many arthropod groups). Proper studies, on other animals, are necessaries to apply efficiently the corridor strategy.

The objective of this study is to assess the effectiveness of corridors to enhancing shrew's occurrence on urban landscapes which are dominated by domestic gardens.

Domestic gardens (or private gardens) are small but numerous green spaces associated with small houses, they can represent more than 1/3 of some cities surfaces (Mathieu et al. 2007; Vergnes et al. 2012). Domestic gardens are generally planted with hedgerows, lawn and trees (Loram et al. 2007). In urban landscapes, they represent a major component of green spaces (Gill et al. 2008; Davies et al. 2009) and especially of woody habitats (Doody et al. 2010). Goddard et al. (2010) highlighted the conservation potential of domestic gardens and the necessity to study them in a landscape ecology framework. Nevertheless, studies that considered the relationship between gardens and other green spaces are still scare (Vergnes et al. 2012).

We focussed our study on shrews (Soricomorpha: Soricidae), because these organisms are highly sensitive to landscape modifications and particularly fragmentation (Pearce and Venier 2005; Michel et al. 2006; Rico et al. 2007; Vuilleumier and Fontanillas 2007; Wang and Grimm 2007). Moreover, corridors have positive effects on shrews in other landscapes (Constantine et al. 2005; Michel et al. 2007; Vuilleumier and Fontanillas 2007), but their functioning in urban landscapes had never been documented before. As they occupied an intermediary place in the food chain, shrews play an important role in many ecosystems. They are the prey of many predators like owls (Balloux et al. 1998) and cats in urban areas (Woods et al. 2003) and also important predators of soil invertebrates (Churchfield 1982; Scheu 2002). Their voracious habits can provide the control of pests in gardens (Churchfield 1990). Despite these advantages, the effect of fragmentation on shrews is still understudied (reviewed by Mortelliti et al. 2010). Finally, few studies have been focused on shrews in cities (Yalden et al. 1973; Dickman and Doncaster 1987; Churchfield 1990; Tikhonova et al. 2009) and the factors controlling their distribution need further investigations.

According to McGarigal and Cushman (2002), the effect of landscape can be studied using two empirical approaches: (i) manipulative, where the experimenter physically manipulates the attribute of the environment and (ii) mensurative, where the experimenter "simply" measures the environment. Many authors encourage the settlement of manipulative experiments in an urban context (Felson and Pickett 2005; Pickett and Cadenasso 2008) but the difficulties inherent to their realization (numerous private ownership to deal with) could explain their rarity. In order to analyse the effectiveness of corridors, we set up a mensurative field experiment which involved a strict sampling. Indeed, we simultaneously trapped shrews in woodlots considered as a source, in woody corridors linked to these woodlots, and in domestic gardens connected to or disconnected from these corridors. According to many authors recommendations (Beier and Noss 1998 ; McGarigal and Cushman 2002; Gilbert-Norton et al. 2010), we took into account many precautions like the presence of a control treatment with replication and the control of the distance to the woodlots. With this strict sampling design, we indirectly analysed the dispersal of shrews (Jacobson and Peres-Neto 2010). As proposed by (Goddard et al. 2010), we explored the factors that could determine their distribution in urban landscapes at both landscape and local scale. We tested three general hypotheses proposed by Mortelliti and Boitani (2009):

(1) the 'landscape hypothesis' where we investigated the effects of spatial configuration of gardens and the effect of the corridor.

(2) the 'local hypothesis' where we investigated the effect of habitat characteristics of the garden like vegetation (Dickman 1987; Dickman and Doncaster 1987; Mortelliti and Boitani 2009) or food availability (Churchfield 1982).

(3) the 'competition hypothesis' where we investigated the effects of interspecific competition. This process has been identified as another major determinant factor of shrew distribution (Ellenbroek 1980; Barnard and Brown 1981)

Using generalized linear model (GLM) with an information theoretic approach (Burnham and Anderson 2002), we compared the importance of the three hypotheses on the occurrence of the different species of shrews in gardens. Finally, we discussed the effectiveness of urban corridors and suggested how they could be a useful tool for sustainable urban planning.

Methods

Study area and sampling design

The study took place in the suburbs of Paris (capital of Paris regions, 48°51′24. 30″N and 2°20′34. 46″E, see Fig. 1). The Paris region is one of the most densely populated regions in Europe with 971 inhabitants/km^2 (INSEE sources, French National Institute for Statistic and Economic Studies). The suburbs of Paris are mainly composed of urban landscapes. Most of them are composed of small houses with domestic gardens and separated by roads. In this urban form, the density is about 2,000–5,000 inhabitants/km^2 (INSEE sources).

To asses the effectiveness of woody corridor to enhancing the occurrence of shrews, we set up a measurative sampling design with a strict landscape configuration on four sites (McGarigal and Cushman 2002). Each site was composed of a unique woodlot in 1.5 km radius (a remnant forest or an urban park up to 150 ha), a woody corridor (a liner strip of wood like road verge between 20 and 50 m wide) physically connected to
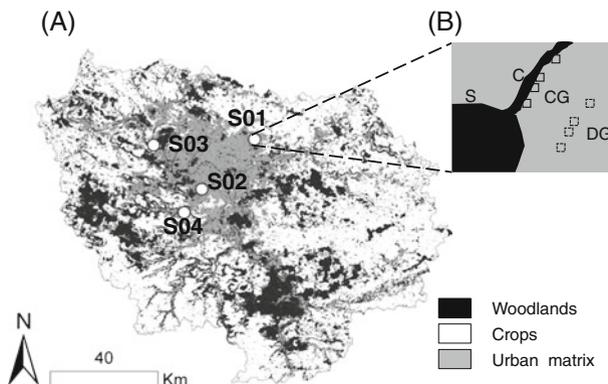

Fig. 1 Map of the sampling sites with land-uses class (**a**) and scheme of the sampling design (**b**) used to analyse corridor effects on shrew occurrences in gardens. The urban matrix is composed of detached houses with garden and streets. CG = gardens connected to or close to the woodlot. DG = disconnected gardens. C = corridor. S = woodlots considered as sources

the woodlot and domestic gardens planted with hedgerows, lawn and trees (Loram et al. 2007). Indeed the woodlots and the corridors are located in suburban landscapes, rounded by many detached houses with domestic gardens. Within those gardens, we selected four gardens close or connected to the corridor and four gardens located at more than 300 m from the corridor. All selected gardens were planted with hedgerows. Overall, we sampled 16 connected gardens, 16 disconnected gardens, four corridors and four woodlots.

Site 01 (here after S01) is located in the north of Paris, the woodlot is a remnant forest and the corridor is a woody riverway. Site 02 (here after S02) is located in the south of Paris; the woodlot is an urban park and the corridor an urban greenway. Site 03 (here after S03) is located in the west of Paris; the woodlot is a forest and the corridor a disused woody railway edge Site 04 (here after S04) is also located in the south of Paris; the woodlot is a remnant forest and the corridor a woody road verge.

In all these sites, the dominant vegetation of the woodlots and corridors is mixed deciduous and is dominated by native oaks (Quercus spp.). The lower strata of vegetation are mainly composed of *Rubus fructicosus* and *Hedera sp*.

Model species

Shrews (Soricomorpha: Soricidae) are small (usually less than five centimetres) and ground-dwelling mammals which lives in leaf litter and dense vegetation (Churchfield 1990). Shrews are mainly nocturnal and move by walking on the ground or by swimming on few kilometres (Hanski 1986). Many shrews are extremely territorial (Churchfield 1990). Their home range are sex and age dependent and varied from 75 m^2 to 1850 m^2 (Churchfield 1990). Actually, they use a reduced foraging area (30 m radius for *S. araneus*) where they spend 95 % of their time and an exploratory area (or "familiar area") of 180–240 m radius (Shchipanov et al. 2011). Dispersal is a key process for many species which shows an almost obligatory exploratory behaviour (Shchipanov et al. 2011). Long distance movements outside their territories (a mean of 170 m) or excursions were frequently observed for *Sorex araneus* (Shchipanov et al. 2008) and they could cover an area of 670–17460 m^2 (Hlyap et al. 1970 in Shchipanov et al. 2008). Moreover many species are organised in metapopulations (i.e. group of spatially separated subpopulations) (Hanski 1989; Peltonen and Hanski 1991). A modeling study showed that the dispersal distance between subpopulations seems limited to 15 km and that they actively used corridors (Vuilleumier and Fontanillas 2007).

Shrew trapping

Shrews were trapped with unbait pitfall traps filled with a preservative solution (for more details, see de Redon et al. (2010)). A line of six traps were place in each woodlot. Within each corridor, we defined three stations of four traps at different distances from the woodlot: 10 m, 80 m and 200 m for site S01; 30 m, 150 m and 300 m for site S03; 30 m, 80 m and 350 m for sites S02 and also S04. Two traps per garden were placed under hedgerows. Overall, 135 traps were laid for this study.

Shrews were trapped from May to June 2009 and from June to July 2009 representing a total of 8100 trapnights. As the two periods showed no statistical difference in shrew abundances, we combined them for analysis (Wilcoxon paired test. $V=397.5$; $P=0.2981$; $n=32$). Our trapping method does not allow the capture of live individuals, so we could not mark and release individuals.

Environmental variables

At the landscape scale, we measured the connectivity of gardens with the nearest Euclidian distance to the edge of the corridors (Dcor) and also the nearest Euclidian distance to the woodlots (Dwood). Both variables have been recorded with ArcGis 9.3 (ESRI).

At the local scale, we measured the area of the garden (Area). We also measured many variables of the vegetation and of management practices in gardens. We recorded the vegetation cover under hedgerows (Veg) in a 4×4 m square around each trap and ordered into one of four categories (0–25 %, 26–50 %, 51–75 %, 76–100 %). We also recorded the height of hedgerow in meters (Hrow_height) ordered in four categories (<2; 2.1–3, 3.1–4 >4.1), the width of hedgerow in meters (Hrow_widht) ordered in three categories (<1, 1.1–2.5, >2.6). Management practices were determined by a questionnaire given to the owners of gardens and personal observations. We considered hedgerow cutting per year (Hrow_cut) scored 0 or 1, frequency of lawn cutting per year (Lawn_cut) ordered in three categories (<5, 6–9, >10), the cutting of trees (Tree_cut) scored 0 or 1, the number of trees taller than 2 m excluding the sampled hedgerows (No. Tree) ordered in three categories (0–4, 5–9, >9), the presence of compost (Compost) scored 0 or 1, the presence of dead woods in garden (scored 0 or 1), the use of herbicides in garden (Herbicide, scored 0 or 1. Use of insecticide (Insecticide) scored 0 or 1 and the use of fertilizer (Fertilizer) scored 0 to 1. With the aim to avoid to avoid over parameterization and multi colinearity we performed a Principal Component Analysis (PCA) which allow find a small number of linear combination of non correlated variable so as to capture most of the variation in the data frame as the whole (Crawley 2009). We summarized the 12 variables in two composite variables (see Online resource 1 for details). The first principal component (here after named PC_veg) explained 28.46 % of the total inertia and represent a gradient of increasing vegetation cover under hedgerows. The second principal component (here after name PC_manage) explained 20.49 % of the total inertia and could be interpreted as a management gradient according to strong loading of these variables (see Online resource 1 for details).

Finally, food availability was assessed using the number of carabids (Carab) captured in the same devices as shrews. These insects are a main component of the shrew's diet (Churchfield 1982; Dickman 1988). The number of individuals captured is established as a reliable estimate for the abundance of carabids (Tyler 2008).

Data analysis

We analysed the binary response (occurrence) of shrews in gardens using generalised linear models (GLM) specifying a binomial error according to the nature of our data (Crawley 2009). Within the framework of the three broad hypotheses defined in the introduction, we computed all the possible combination of uncorrelated variables (simple, additive effect and interaction effects in a limit of four parameters, see Online resource 2 for details). The list of all the models computed is given in the Online resource 3.

We log-transformed all variables associated with distance, area and abundance of carabids prior to analyses.

The first broad hypothesis, referred to as 'landscape hypothesis', was that occurrence of shrews would be affected by landscape process. We considered the spatial configuration of gardens (Dcor and Dwood) but also the abundance of shrews in corridors and woodlots. As many organisms, the dispersal rate of shrews is correlated with the density of individuals present in potential sources like woodlots and corridors (Wang and Grimm 2007).

The second broad hypothesis, referred to as 'local hypothesis', was that the characteristic of gardens at the local habitat scale, the area of gardens and the food availability would explain the occurrences in gardens. Thus, predictor covariates included PC_veg, PC_manage, Area and Carab.

The last broad hypothesis, called 'competition hypothesis', was that the occurrence in gardens would be affected by interspecific competition or co-occurrence (Ellenbroek 1980; Mortelliti and Boitani 2009) and thus the abundance of the other species captured in the same garden was used as covariate.

Using an information-theoretic approach for model selection, models were ranked according to their AIC_C (Akaike Information Criterion Corrected for small sample size). We also calculated the ΔAIC_C and Akaike weights (Burnham and Anderson 2002). The ten best ranked models for each species and for the total shrew occurrence are represented in Table 2. The analyses were computed with R 2.7.0 software (R Development Core Team 2011).

Results

Distribution of shrews in the landscape

Overall, we captured 97 shrews belonging to three species (capture rate of 0.011 individuals per trapnight). Shrews occurred in 25 % of the gardens (Table 1). Nevertheless, they occurred in 43 % of connected gardens ($n=16$) and in only 6.3 % of disconnected gardens ($n=16$). The Millet's shrew *Sorex coronatus* and the pygmy shrew *Sorex minutus* were found in all woodlots but only in 25 % of corridors. These two species occurred only in connected

Table 1 Abundance, mean and the standard deviation (SD) of the shrews captured in the different type of green spaces: woodlots, corridors and gardens in the four sites (S01 to S04).

Type of green space	Site	*Crocidura russula*	*Sorex coronatus*	*Sorex minutus*	*Shrews*
Woodlots ($n=4$)	S01	0	10	18	28
	S02	0	3	2	5
	S03	1	11	1	13
	S04	0	1	1	2
	Total	1	25	22	48
	Mean (SD)	0.25 (0.25)	6.25 (2.50)	5.5 (4.17)	12 (5.82)
Corridors ($n=4$)	S01	0	6	3	9
	S02	4	0	0	4
	S03	1	0	0	1
	S04	2	0	0	2
	Total	7	6	3	16
	Mean (SD)	1.75 (0.85)	1.50 (1.50)	0.75 (0.75)	4 (1.78)
Gardens ($n=32$)	S01	2	2	9	13
	S02	0	0	1	1
	S03	3	0	0	3
	S04	2	0	0	2
	Total	7	2	10	19
	Mean (SD)	1.67 (0.63)	0.50 (0.50)	2.50 (2.17)	4.75 (2.78)

gardens ($n=16$) (respectively 12.5 % and 18.8 %). The greater white-toothed shrew *Crocidura russula* was found in 25 % of woodlots, 75 % of corridors, 31 % of connected gardens and 6 % of disconnected gardens. The results for carabids were detailed in its own study (Vergnes et al. 2012).

Landscape scale, local habitat scale and competition effects on occurrence

The ten first ranked models from GLM were shown in Table 2. Models with a ΔAICC >2 are considered less relevant (Burnham and Anderson 2002).

For shrews overall, the three first ranked models had close AIC_C and involved the spatial configuration. The distance to the corridor was present in the three first ranked models, with a negative effect on the occurrence (Table 2). Only three models from the local hypothesis were present in the top ten but with a ΔAIC$_C$ >2.

For *C. russula*, the first two models involved spatial configuration and had really close AICc. The first model expressed the probability of occurrence of *C. russula* as a negative function of the distance to the corridor and the second combined the distance to the woodlot and the distance to the corridor. The fourth model expressed the occurrence of *C.russula* as a positive function of the occurrence of *S. minutus*. Models from the 'habitat hypothesis' were poorly ranked.

The first ranked model for *S. coronatus* involved the local hypothesis. It expressed the probability of occurrence as a negative function of PC_manage, describing a gradient of increasing management. The other models have considerably less support.

The first ranked model for *S. minutus* was associated with the landscape hypothesis and expressed the probability of the occurrence as a negative function of the distance to the woodlot and to the corridor. The second and the third ranked models were associated with the 'local hypothesis'. They expressed respectively the occurrence as a positive function of Carab, and an additive effect between PC_veg and PC_manage. The third model and the fifth model were also associated with 'landscape hypothesis' and showed a negative effect of Dwood and the same effect plus an additive positif effect of ab_min.

Discussion

Our results highlighted a strong effect of the landscape scale on shrew occurrences in gardens and a clear positive effect of corridors, stronger for *C. russula*. In a lesser extent, the local scale affected the occurrence through the management of gardens and the structure of vegetation.

Distribution of shrews on urban landscape

In term of diversity, we captured three species of shrews in the studied sites when actually four species are present in the whole region (Mitchell-Jones et al. 1999). The total abundance was relatively similar to those found in other studies conducted in agricultural or more natural areas. Indeed, Mortelliti and Boitani (2009) get a capture rate of 0.016 individuals per trapnight in farming landscapes, Michel et al. (2006) get 0.006 individuals per trapnight in hedgerow networks located on farming landscapes and here, we obtain 0.011 individuals per trapnight. In our study, the three shrew species did not behave similarly. As in Yalden et al. (1973), we observed that *C. russula*, a facultative synanthrop species (or resistant to human activities) sometimes called the house shrew (Churchfield 1990), occurred mainly in

Table 2 Summary of the top ten ranked generalised linear models of shrew occurrences

Species	Model	Hypothesis	AIC$_C$ weight	ΔAIC$_C$	β (se)
Shrews	Dcor X Dwood	land	0.26	0.00	−0.38 (0.19)
	Dcor + Dwood	land	0.23	0.24	−0.98 (0.51) −1.55 (0.99)
	Dcor	land	0.16	0.93	−0.8 (0.43)
	ab_shrew_wood + Dcor	land	0.10	1.93	0.04 (0.03) −1.00 (0.48)
	ab_shrew_cor + Dcor	land	0.07	2.64	0.13 (0.13) −0.95 (0.47)
	Dwood	land	0.05	3.21	−1.34 (0.85)
	PC_veg	hab	0.04	3.61	0.98 (0.69)
	Area	hab	0.03	4.29	1.920 (1.649)
	PC_veg + Carab	hab	0.03	4.65	1.27 (0.767) 1.35 (1.072
	ab_wood	land	0.02	4.91	0.02 (0.02)
Crocidura russula	Dcor X Dwood	land	0.28	0.00	−0.31 (0.20)
	Dcor	land	0.28	0.01	−0.80 (0.48)
	ab_rus_cor + Dcor	land	0.21	0.58	−0.51 (0.40) −0.95 (0.51)
	gard_min	compet	0.13	0.89	0.58 (1.00)
	Dcor + Dwood	land	0.07	1.41	−0.52 (1.04) −0.80 (0.47)
	ab_rus_cor	land	0.03	1.79	−0.40 (0.35)
	PC_veg	hab	0.01	1.89	0.92 (0.76)
	PC_veg X PC_manage	hab	0.00	1.93	−1.59 (1.38)
	PC_manage	hab	0.00	1.94	0.09 (0.82)
	Area	hab	0.00	2.06	2.57 (1.887)
Sorex coronatus	PC_manage	hab	0.74	0.00	−0.35 (1.197)
	gard_rus	compet	0.20	2.61	2.88 (2.03)
	PC_veg + PC_manage	hab	0.05	2.90	1.60 (2.68) 5.44 (3.54)
	Dcor	land	0.01	3.00	−0.26 (065)
	PC_veg	hab	0.00	3.08	0.86 (2.71)
	PC_veg X PC_manage	hab	0.00	3.52	−0.82 (2.046)
	Area	hab	0.00	3.80	0.89 (1.112)
	PC_veg + Area	hab	0.00	5.66	−0.44 (1.25) 1.04 (2.61)
Sorex minutus	Dcor + Dwood	land	0.33	0.00	−1.67 (1.38) −4.79 (2.24)
	Carab	hab	0.28	0.33	4.87 (1.14)
	Dwood	land	0.22	0.45	−4.62 (1.35)
	PC_veg + PC_manage	hab	0.10	1.58	−0.15 (0.97) 0.02 (0.97)
	ab_min_wood + Dwood	land	0.04	1.85	0.08 (0.07) −2.75 (1.46)
	PC_manage + Carab	hab	0.01	2.01	−1.31 (1.35) 5.25 (2.56)
	PC_veg + Carab	hab	0.00	2.92	0.51 (1.30) 4.27 (2.05)
	Area + Carab	hab	0.00	2.95	0.94 (2.73) 4.21 (2.00)
	ab_min_wood	land	0.00	4.26	4.10 (1.95)
	Dcor X Dwood	land	0.00	4.27	−0.46 (0.40)

Models are ranked according to ΔAIC$_C$ (see Methods for details). Estimates (β) with standard error were specified for each covariate. For landscape hypothesis (land), we used the distance to the corridor (Dcor) the distance to the woodlot (Dwood), abundance of *C. russula* (ab_rus_), the abundance of *Sorex coronatus* (ab_coro_), the abundance of *Sorex minutus* (ab_min_) in woodlot when followed by wood and in corridor when followed by cor. For local habitat hypothesis (hab), we used the log transformed area of garden (Area), the vegetation cover under hedgerow (PC_veg), the management of gardens (PC_manage) and the log transformed abundance of carabids (Carab). For competition hypothesis (compet), we used for each species the abundance of the other species (gard_min, gard_rus and gard_coro). Simple interactions between covariates are signed by X

gardens whereas *S. coronatus* and *S. minutus* were more present in woodlots and corridors and more rarely in gardens. Surprisingly, *S. minutus* was the most abundant species whereas some authors considered it an exoanthropic species (or rare in cities) (Tikhonova et al. 2009). Furthermore, when considering only the connected gardens, we highlighted a higher occurrence of the *Sorex* species than in other studies (less than 5 % of occurence in Dickman (1987), 17 % of *S. araneus* in Dickman and Doncaster (1987) but with only six gardens).

Effects of local habitat quality and interspecific competition

The three shrew species showed sensitivity to the local habitat scale and quite similar response to garden characteristics. As in many studies (Dickman and Doncaster 1987; Mortelliti and Boitani 2009), the structure of the vegetation (measured here above hedgerows of gardens), played an important role in shrew occurrence. Some authors (Dickman and Doncaster 1987; Churchfield 1990) considered that the vegetation affected shrews indirectly and mainly by increasing invertebrate abundances and in a lesser extent by protecting shrews from predators. In domestic gardens, the response of invertebrate abundances is still unclear (Smith et al. 2006) and the abundance of carabids, an important component of the shrew diet (Churchfield 1982), was only well ranked for *S. minutus*. The role of the vegetation in limiting predation seems more plausible as predation by cats (*Felis catus*) was stronger in domestic gardens than in other human dwellings (Woods et al. 2003). Garden management was also important for the occurrence of shrews. The application of chemicals (chemical herbicides or insecticides) could pose a severe risk of secondary poisoning for shrews (Churchfield 1990) and are sometimes applied in gardens in higher concentrations than in agricultural landscapes (Blanchoud et al. 2004).

As in other studies (Mortelliti and Boitani 2009), we observed that many species co-occured in small patches. Interspecific competition is considered as a major determinant factor of shrew distribution (Ellenbroek 1980; Barnard and Brown 1981) but was not observed here. Some patterns as good local habitat quality and abundant resources could relax the interspecific competition in such small patches and facilitate the co-occurrence of several species in gardens (Mortelliti and Boitani 2009).

Importance of spatial configuration and corridors on shrew occurrences

We highlighted that spatial configuration of gardens, mainly through the presence of the corridor, were major factors explaining the distribution of shrews in urban landscapes. As a consequence, shrews were not widely distributed in gardens which revealed the importance of corridors in urban landscapes.

The effects of the spatial configuration on shrews are not clear and depend of the landscape studied. Shrews are sometimes considered as "good dispersers" for small mammals (Churchfield 1990) and some studies showed that the properties of agricultural landscapes did not affect their distribution (Dickman and Doncaster 1987) or in a lesser extent than local variables (Michel et al. 2007; Mortelliti and Boitani 2009). In addition, shrews are known to colonize islands located a few kilometres away from mainland, by swimming or walking on the ice (Peltonen and Hanski 1991). In our study, the spatial configuration, and more precisely the distance to the corridor, clearly affected the distribution of shrews. The importance of landscape configuration in cities has been demonstrated before for small mammals (Baker and Harris 2007) but the positive effect of corridor in such context is unheard of. Contrary to rural landscapes, the urban matrix is not composed of crops or open areas but of roads and buildings. For many species, these structures were associated with a

high mortality risk during dispersal and are considered as strong barriers (Coffin 2007), particularly for small mammals (Rico et al. 2007). Moreover, we showed that without corridors, shrews could not reach domestic gardens including those located less than 100 m away from the woodlots, a distance in the exploratory range of individuals (Churchfield 1990; Shchipanov et al. 2011) and even more of subpopulations (15 km in Vuilleumier and Fontanillas 2007). Many authors (Dickman and Doncaster 1987; Peltonen and Hanski 1991) suggest that *C. russula*, a facultative synanthrop species could be able to disperse across urban landscapes and exploit more efficiently the urban context than *Sorex* species. However, Vuilleumier and Fontanillas (2007) showed that *C. russula* is highly sensitive to fragmentation. These contradictory results could illustrate the difficulty to measure properly the effects of landscapes on organisms (McGarigal and Cushman 2002) and especially fragmentation (Fahrig 2003). Here, we clearly demonstrated that without corridors, the functioning of shrews as metapopulations (Peltonen and Hanski 1991) could be compromised in such fragmented urban local habitat.

Sampling design: Benefits and limits

With our mensurative experiment we highlighted that corridors enhanced the occurrence of shrews in gardens. However, the role of corridors—habitat, conduit or both (Hess and Fischer 2001)—and the origin of the individuals captured in gardens are two questions that remain unanswered.

The shrews captured could be vagrants coming from woodlots, corridors or both. As gardens were located within the explanatory distance of many shrews and especially *Sorex* species (Shchipanov et al. 2008, 2011), these individuals may have come directly from woodlots through corridors. In that case, corridors would have a 'conduit function' (Hess and Fischer 2001) and would have to be associated with woodlots in green frameworks to be fully efficient.

Shrews would also be able to live and reproduce in corridors. Corridors would be able to support their own populations and would have the 'habitat function' (Hess and Fischer 2001). Without further investigations, no final conclusion could be made but the importance of the density of shrews in the woodlots argues for the 'conduit function'.

The individuals captured could be a part of populations living in the gardens. For *C. russula*, the reproductive unit necessary for successful colonisation could be very small (Vogel 1999) and their home range varied from 75 to 395 m^2 (Churchfield 1990). The home range of *Sorex* species and especially *S. minutus* are larger (900–1.850 m^2) (Churchfield 1990) but their foraging area (95 % of the animal unit activity) can be much smaller (30 m radius) (Shchipanov et al. 2011). In our study, the surface of gardens was highly variable (mean (se)=516 m^2 (469)) but many gardens were large enough to contain the home range of many species. Moreover, shrews could use the surrounding gardens in breeding season to enlarge their home range as observed in agricultural matrix (Wang and Grimm 2007).

More studies using direct tracking, recapture techniques or genetics (Storfer et al. 2010) are needed to clarify if gardens and corridors could be considered as local habitat and in that case if corridors respond to the source function (Hess and Fischer 2001).

Conclusions

We highlighted the positive role of corridors in the occurrence of shrews in domestic gardens. Without corridors, the classical idea of shrew populations as metapopulations with

colonisation-extinction processes (Peltonen and Hanski 1991) would be compromised in such fragmented urban habitat. Many studies found a weaker effect of landscape on shrews. Our sampling design—repeated, at a large scale and comparative—was particularly efficient to measure the effect of landscape connectivity and to distinguish landscape from local habitat effects (McGarigal and Cushman 2002; Mortelliti and Boitani 2009; Mortelliti et al. 2010).

The effectiveness of corridors and especially woody corridors, in increasing animal and plant movement has been demonstrated in numerous landscapes (Bryant 2006; Gilbert-Norton et al. 2010). In cities, corridors have important recreational and pedagogical values (Savard et al. 2000). Our results comfort their ecological role in the conservation of ordinary biodiversity in urban domestic gardens, a major component of urban green spaces, and argue for their promotion in sustainable urban planning (Ahern 2007).

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